DEVELOPMENT OF NUTRIENT SUBMODULES FOR USE IN THE GRIDDED SURFACE SUBSURFACE HYDROLOGIC ANALYSIS (GSSHA) DISTRIBUTED WATERSHED MODEL[1]

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ABSTRACT: A primary water quality problem caused by non-point source pollution (NPSP) is eutrophication, from excess nutrients in receiving water bodies. The control of nutrients arising from NPSP is difficult because the source areas can be hard to identify and typical treatment methods are infeasible due to the distributed nature of the pollutants. It may be possible to reduce nutrient related water quality problems through the restoration of highly disturbed watersheds with best management practices (BMPs). While restoration attempts may provide significant returns, they can be costly to implement and often are met with resistance in agricultural communities. Extending model results beyond the range of calibration to model future conditions such as for restoration scenarios requires the use of physically-based models that include the important processes that generate streamflow and material transport, uptake, loss, transformation, and recycling of nutrients and other material. The research and development objectives of the U.S. Army Engineer Research and Development Center (ERDC) in Vicksburg, Mississippi, are to develop a watershed assessment and management model to simulate transport, uptake, loss, transformation, and recycling of nutrients such as nitrogen and phosphorus and associated material such as sediment and organic matter. In this study we will discuss current efforts at the ERDC's Environmental Laboratory to develop a state-of-the-art watershed water quality model.
(KEY TERMS: watershed; nutrients; water quality; distributed; GSSHA.)

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INTRODUCTION

Multiple riverine, estuarine, watershed, and sub-surface flow models are being modified at the ERDC to address issues of environmental concern. Several approaches are either ongoing or proposed to accomplish this task. Other focus areas of the System-Wide Water Resources Research Program (SWWRP) are addressing the task of interfacing models of differing spatial dimensions in order to facilitate system-wide capability. To have a full system-wide water quality and contaminant capability in SWWRP, the different hydrologic and hydraulic engines must utilize a common water quality and contaminant approach to prevent the arbitrary portioning of constituents. The goal of this research and development effort is to upgrade existing hydrologic and hydraulic models (i.e., water engines) using a common water quality approach to facilitate their linkage and application on a system-wide basis. ERDC engineers and scientists are working with universities, private companies, and other government agencies to develop better physical representations of nutrient fate and transport algorithms. These algorithms will be incorporated into a common library for use by a variety of water engines. Work also is ongoing to develop an efficient means to populate models and to store model parameters and results for future use. Links to national, regional, and local databases are being made such that development of distributed models becomes less cumbersome. Storage of model input data, parameter estimations, and model output will be done in a way that facilitates their use on future studies. The goal is to import

all U.S. Army Corps of Engineers numerical studies in an effort to reduce model setup time and to enhance decision-support efforts.

The model developers recognize and appreciate previous development efforts in the watershed water quality technical area of models such as SWAT, AGNPS, and HSPF. The ERDC development effort differs from those models in that nutrient submodules are being developed to represent physically, as much as possible, the fate and transport of nutrients within and between the soils, overland, and channel systems. These nutrient modules are being integrated into complex and physically based hydraulic and hydrologic modeling systems to achieve detailed transport between and across the various hydrologic regimes. While models such as SWAT and AGNPS simulate detailed fate processes, they tend to be oriented more toward agricultural needs, and the transport schemes tend to lump the processes so that detailed analysis of BMPs within a subarea cannot be readily accessed. Models such as HSPF enable modeling of the whole hydrologic cycle; however, the process descriptions tend to be relatively simplistic, which makes it difficult to transfer the model to geographic areas for which large datasets may not be available.

The library of algorithms will be able to accommodate a multispecies, multiphase, multireaction system; will include both fast (equilibrium-based) and slow (nonequilibrium-based or rate-based) reactions; will be easily extensible to new reaction pathways; will include common nutrient and contaminant packages as well as geochemistry; and will have a simple, well defined data interface and calling procedure, making them portable. The modules will be developed to be independent of data structure, thus facilitating their integration into a wide range of modeling systems.

DEVELOPMENT APPROACH

In fiscal year 2005, two modeling systems were being linked with the current version of SWWRP-NSM. Those two modeling systems are the Gridded Surface Subsurface Hydrologic Analysis (GSSHA) model and the Hydrologic Engineer Center River Analysis System (HEC-RAS). In this study we focus on the efforts under way to develop a GSSHA distributed watershed water quality model.

GSSHA Model

Methodologies within GSSHA that encompass the hydrologic cycle (Downer and Ogden, 2002) include (1) precipitation distribution, (2) interception, (3) infiltration, (4) evaporation and evapotranspiration, (5) overland flow, (6) channel flow, (7) ground water flow, (8) coupling of saturated zone and unsaturated zone, and (9) soil erosion and sediment routing.

Rainfall is a required input within any hydrologic model. Within GSSHA, rainfall may be input as spatially and temporally uniform at a specified rate for a specified duration for a single event, or rainfall may be input as spatially and temporally varying for any number of rainfall events. The rainfall interpolation techniques that are available for spatially varied rainfall are the inverse distance squared method and the Theissen polygon method. The Theissen polygon method is simply a nearest neighbor approach, while the inverse distance squared method produces smooth fields based on the assumption that the influence of a measured value decreases with the distance from the point of measurement squared. NEXRAD precipitation estimates can be used in GSSHA by formatting the data into a GSSHA precipitation file using the RADAR precipitation type card. When using NEXRAD rainfall estimates, GSSHA assigns a rain gage at the center of each radar data pixel. When combined with Theissen polygon rainfall interpolation, this reproduces the original radar pixels.

The interception of rainfall by the vegetation is modeled in GSSHA using the two-parameter method published by Gray (1970). An initial quantity of rainfall (mm) entirely intercepted by foliage and a storage capacity are specified within the model for each land use type. As the plant soil modules are developed for SWWRP-NSM, a more robust accounting for interception will be available.

The evaporation and evapotranspiration (ET) models incorporated in GSSHA allow calculation of the loss of soil water to the atmosphere, improving the determination of soil moistures. Two evapotranspiration options are bare ground evaporation from the land surface using the formulation suggested by Deardorff (1978) and evapotranspiration from a vegetated land surface using the Penman-Monteith equation (Monteith, 1965, 1981). Variants of these two representations are widely used in land surface schemes of climate and distributed hydrologic models (e.g., Beven, 1979; Dickinson *et al.,* 1986).

GSSHA allows water to be ponded on overland flow plane cells that will infiltrate into the soil as conditions permit. Infiltration depends on soil hydraulic properties and antecedent moisture conditions, which may be affected by previous rainfall, run-on, ET, and

the location of the water table. In GSSHA, the unsaturated zone that controls infiltration may be simulated with a one-dimensional (1-D) formulation of Richards equation (RE), which simulates infiltration, ET, and soil moisture movement in an integrated fashion. Infiltration may also be simulated using traditional Hortonian Green and Ampt (GA) approaches, which are simplifications of RE. There are three optional GA-based methods to calculate infiltration for Hortonian basins: traditional GA infiltration, multilayer GA, and Green and Ampt infiltration with redistribution (GAR) (Ogden and Saghafian, 1997). The traditional GA and multilayer GA approaches are used for single-event rainfall when there are no significant periods of rainfall hiatus. The GAR approach is used when there are significant breaks in the rainfall or for continuous simulations.

Overland flow in GSSHA employs the diffusive wave approximation in two dimensions (x and y). Flow is routed in two orthogonal directions in each grid cell during each time step. The watershed boundary represents a no-flow boundary for the overland flow routing, and when a grid cell lies on the watershed boundary, flow is not routed across the boundary. Once water enters a "channel" grid cell, the volume of water is added to the channel system and routed to the watershed outlet. The overland flow routine does allow for depression storage, and thus water can pool in a depression until it can build up enough head to overcome the topography, infiltrate into the ground, or evaporate into the air.

GSSHA solves the diffusive wave equation using two-step explicit finite volume schemes to route water for both 1-D channels and 2-D overland flow, where flows are computed based on heads and volumes are updated based on the computed flows. Compared with more sophisticated implicit finite difference and finite element schemes, the algorithm used in GSSHA is relatively simple. The friction slope between one grid cell and its neighbors is calculated as the difference in water-surface elevations divided by the grid size. Compared with the kinematic wave approach, this diffusive wave approach allows GSSHA to route water through pits or depressions and through regions of adverse slope. The Manning formula is used to relate flow depth to discharge. Use of the Manning formula implies that the flow is turbulent and that the roughness does not depend on flow depth. Although simple, the method is powerful because it allows calculations to proceed when only portions of the stream network or watershed are flowing. This is an important attribute, as rainfall may occur on only part of the watershed.

The channel routing scheme was developed to allow water to remain in the channel after channel routing ends and for water to be present in the channel when channel routing begins. Because ground water may discharge to the stream at any time, channel routing is initiated when a minimum amount of water is in the channel network. If the channel routing scheme indicates no flow in the channel, channel routing is halted during periods outside precipitation events. Fluxes between the stream and the ground water are still computed, and adjustments to the stream volumes are made without routing. If ground water discharges to the stream, channel routing will resume but at the ground water time step, which is typically larger than the channel routing time step.

Trescott and Larson (1977) described the solution to the two-dimensional free surface ground water problem and the efficiency of various solvers. Their methods were largely followed in developing this part of the code; exhaustive coverage need not be presented here. The ground water equation is solved by line successive overrelaxation (LSOR) (for example, Tannehill *et al.,* 1997). Trescott and Larson (1977) showed LSOR to be capable of solving a variety of difficult ground water problems, though not necessarily the most efficient method. With LSOR, the two-dimensional problem is linearized by solving by rows or by columns. The user specifies solution by rows or by columns, and the choice is made to align the direction of solution with the principal direction of flow, x or y, (Trescott and Larsen, 1977).

In GSSHA, the saturated and unsaturated zones are linked though boundary conditions. When saturated ground water is simulated, the lower boundary of the unsaturated zone is the saturated ground water surface. Movement of the saturated ground water surface according to the solution of the 2-D saturated flow equations requires a flexible spatial discretization for Richards equation. In extreme cases the ground water table may rise to the soil surface. In this case the unsaturated zone disappears, and only the saturated flow equations are solved. When heads from the 2-D unconfined ground water problem solution are used as the lower boundary condition of the unsaturated ground water problem, the size of the unsaturated zone in each overland flow cell changes with each saturated ground water update. Also, the storage term used in the saturated ground water solution does not account for the water in the unsaturated zone. After solution of the saturated zone, an extra step is required to account for the water that exists in the unsaturated zone.

In order to estimate overland erosion, GSSHA employs an equation based on the work of Kilinc and Richardson (1973). These authors studied the mechanics of overland soil erosion in at Colorado State University Engineering Research Center in a flume that is 1.2 m deep, 1.5 m wide, and 4.9 m long. Their investigation resulted in a sediment transport

equation of uniform flow sheet and rill erosion on bare sandy soil. Dr. Pierre Julien modified the original Kilinc-Richardson equation to expand the applicability of the equation to nonuniform flow with consideration of soil and land use specific factors (i.e., USLE factors K, C, and P) (Julien *et al.,* 1995). The K, C, and P factors are empirical coefficients with the same conceptual meaning as those used in the Universal Soil Loss Equation (Renard *et al.,* 1991). The surface of each grid cell is either eroded or aggraded depending on the quantity of sediment in suspension and the potential sediment transport rates. This determination is made for three grain sizes, sand, silt, and clay. Conservation of mass of sediment determines what amount of sediment entering each grid cell stays in suspension and what amount is deposited. The sediment transport capacity is satisfied by sediments already in suspension, previously deposited sediments, and then sediments in the parent material. If sediments in suspension are unable to satisfy the potential transport rate, the previously deposited sediment is used to satisfy the demand. If there is insufficient previous deposition, the surface is eroded to meet the demand. If the potential sediment transport rates calculated are insufficient to transport the sediment already in suspension within a grid cell, sediment is deposited on the surface (Johnson, 1997).

The present version of GSSHA employs the unit stream power method of Yang's (1973) for routing sand size total load in stream channels. Unit stream power is defined as the product of the average flow velocity, U, and the channel slope S_o. The rate of work done per unit weight of water in transporting sediment is assumed to be directly related to the rate of work available per unit weight of water. Thus, the total sediment concentration or total bed material load must be directly related to the unit stream power. The routing formulation for sand size sediments is limited to trapezoidal channels with a user specified maximum allowable depth of degradation in the channel. The channel bed is assumed to be mobile, and the banks are fixed. Degradation continues and bed load is transported at the rate calculated with the Yang (1973) method until the maximum degradation is reached. During degradation the initial bed width is maintained and degradation is uniform across the width of the bed. If the channel aggrades, the trapezoidal cross-section is filled. If a channel link has aggraded and then degrades, the degradation will occur uniformly over the bottom of the trapezoid until the original bed elevation is restored. Further degradation occurs vertically downward from the initial trapezoid bottom width. If a channel degraded below the original bed elevation begins to aggrade, sediment will accumulate uniformly in the rectangular degraded area below the original bed elevation. Once the bed

aggrades beyond the original bed elevation, the entire width of the trapezoid is filled. In the channels, silt and clay size particles are assumed to be in suspension and are transported as wash load. This treatment implies that the flow is turbulent, and the travel time to the outlet of the catchment is short compared to the settling time such that particles do not settle in the channel network. This assumption, combined with no bank erosion, results in the channels being neither a source nor sink of fines.

SWWRP-NSM Methodology

The water column modules within SWWRP-NSM can account for the following processes (Johnson and Gerald, 2005):

- Partition N and P between dissolved, adsorbed, and solid states.
- Compute chlorophyll *a*.
- Compute algal growth.
- Compute local specific growth rate of algae.
- Compute algal growth limiting factor for light.
- Compute algal growth limiting factor for nutrients.
- Compute local respiration rate of algae.
- Compute local settling rate of algae.
- Compute organic nitrogen by conversion from algal biomass.
- Compute ammonium by mineralization of Org. N. and diffusion of ammonium from streambed sediments.
- Compute nitrite by conversion of NH4 to NO2.
- Compute nitrate by conversion of oxidation of NO2.
- Compute organic phosphorus by conversion of algal biomass.
- Compute inorganic phosphorus by mineralization of organic P and diffusion of inorganic phosphorus from streambed sediments.
- Compute carbonaceous biological oxygen demand (CBOD).
- Compute dissolved oxygen (DO).
- Compute reaeration.

The water column processes have been implemented within both the overland and the channel modules of GSSHA. The reason for including these processes within the overland kinetics modules is because of the capability of the GSSHA overland module to account for wetlands and detention ponds. Most surface runoff processes will be too quick to take advantage of these processes; however, the grid cells designated to be wetlands and/or ponds should be able to take advantage of the full suite of water column kinetics.

The plant/soil kinetics, nitrogen and phosphorus, have been taken primarily from the SWAT (EPIC) model formulation (Neitsch *et al.,* 2002). While the water column kinetic modules have been implemented within various U.S. Army Corps of Engineers (USACE) modeling systems, the plant/soil kinetic modules are still under development and will be implemented in fiscal year 2006.

Nitrogen Cycle

The three major forms of nitrogen in mineral soils are organic nitrogen associated with humus, mineral forms of nitrogen held by soil colloids, and mineral forms of nitrogen in solution. Nitrogen may be added to the soil by fertilizer, manure, or residue application; fixation by symbiotic or nonsymbiotic bacteria; and rain. Nitrogen is removed from the soil by plant uptake, leaching, volatilization, denitrification, and erosion. Figure 1 shows the major components of the nitrogen cycle.

Nitrogen is considered to be an extremely reactive element. The highly reactive nature of nitrogen results from its ability to exist in a number of valence states. The valence state or oxidation state describes the number of electrons orbiting the nucleus of the nitrogen atom relative to the number present in an electronically neutral atom. The valence state will be positive as the atom loses electrons and negative as the atom gains electrons. The ability of nitrogen to vary its valence makes it a highly mobile element. Predicting the movement of nitrogen among the different pools in the soil is critical to the successful management of this element in the environment.

Figure 2 shows five pools of nitrogen in the soil. Two pools are inorganic forms of nitrogen, NH_4^+ and NO_3^-, while the other three pools are organic forms of nitrogen. Fresh organic N is associated with crop residue and microbial biomass, while the active and stable organic N pools are associated with the soil humus. The organic nitrogen associated with humus is partitioned into two pools to account for the variation in availability of humic substances to mineralization.

Decomposition is the breakdown of fresh organic residue into simpler organic components. Mineralization is the microbial conversion of organic plant-unavailable nitrogen to inorganic plant-available nitrogen. Immobilization is the microbial conversion of plant available inorganic soil nitrogen to plant-unavailable organic nitrogen. Bacteria decompose organic material to obtain energy for growth processes. Plant residue is broken down into glucose, which is then converted to energy
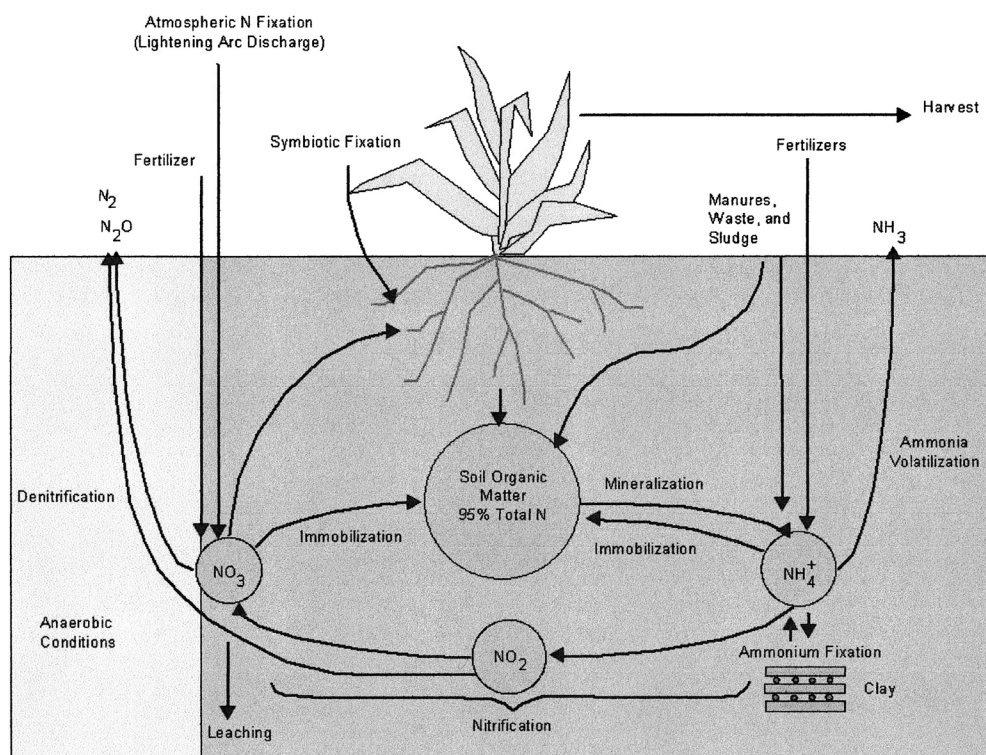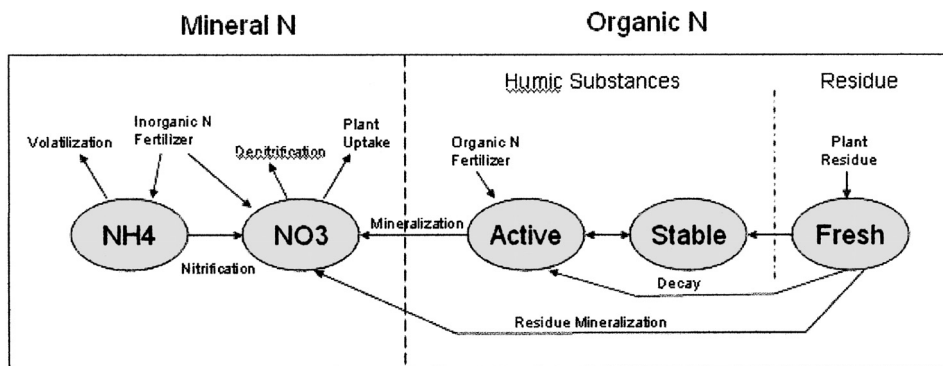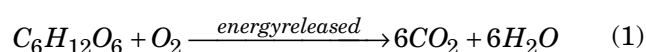

Figure 1. Nitrogen Cycle.

Figure 2. Soil Nitrogen Pools and Processes That Move Nitrogen In and Out of Pools.

$$C_6H_{12}O_6 + O_2 \xrightarrow{\;energy\,released\;} 6CO_2 + 6H_2O \qquad (1)$$

The energy released by the conversion of glucose to carbon dioxide and water is used for various cell processes, including protein synthesis. Protein synthesis requires nitrogen. If the residue from which the glucose is obtained contains enough nitrogen, the bacteria will use nitrogen from the organic material to meet the demand for protein synthesis. If the nitrogen content of the residue is too low to meet the bacterial demand for nitrogen, the bacteria will use ammonium and nitrate from the soil solution to meet their needs. If the nitrogen content of the residue exceeds the bacterial demand for nitrogen, the bacteria will release the excess nitrogen into the soil solution as ammonium. The following shows the general relationship between C:N ratio and mineralization/immobilization:

C:N > 30:1
Immobilization occurs; a net decrease in soil ammonia and nitrate results.

20:1 <= C:N <= 30:1
No net change expected; immobilization and mineralization processes are at equilibrium.

C:N < 20:1
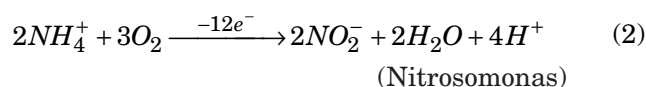Mineralization occurs; a net gain in soil ammonia and nitrate occurs.

Nitrogen is allowed to move between the active and stable organic pools in the humus fraction. The amount of nitrogen transferred from one pool to the other is calculated as a function of a rate constant amount of nitrogen in the active organic pool, a fraction of humic nitrogen in the active organic pool, and the amount of nitrogen in the stable organic pool. When the amount of nitrogen to be transferred is positive, nitrogen is moving from the active organic pool to the stable organic pool. When the amount of nitrogen to be transferred is negative, nitrogen is moving from the stable organic pool to the active organic pool. Mineralization from the humic active organic N pool is calculated as a function of a rate constant for mineralization of the humus active organic nutrients, the nutrient cycling temperature factor, the nutrient cycling water factor, and the amount of nitrogen in the active organic pool. Nitrogen mineralized from the humus active organic pool is added to the nitrate pool in the layer.
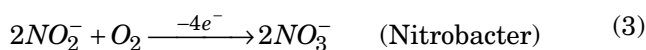
Decomposition and mineralization of the fresh organic nitrogen pool will be allowed only in the first soil layer. Decomposition and mineralization are controlled by a decay rate constant that is updated daily. The decay rate constant is calculated as a function of the C:N ratio and the C:P ratio of the residue, temperature, and soil water content. The decay rate constant defines the fraction of residue that is decomposed and is a function of the rate constant for mineralization of the residue of fresh organic nutrients, the nutrient cycling residue composition factor, the nutrient cycling temperature factor, and the nutrient cycling water factor. The nutrient cycling residue composition factor is a factor of the C:N ratio of the residue in the soil layer and the C:P ratio of the residue in the soil layer. Mineralization from the residue of fresh organic N is a function of the residue decay rate constant and the nitrogen in the fresh organic pool. Nitrogen mineralized from the fresh organic pool will be added to the nitrate pool in the layer. Decomposition from the residue fresh organic N pool is a function of the residue decay rate constant and the nitrogen in the fresh organic pool.

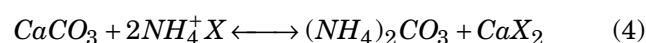Nitrification is a two-step bacterial oxidation of NH_4^+ and NO_3^-.

Step 1:

$$2NH_4^+ + 3O_2 \xrightarrow{-12e^-} 2NO_2^- + 2H_2O + 4H^+ \qquad (2)$$
$$\text{(Nitrosomonas)}$$

Step 2:

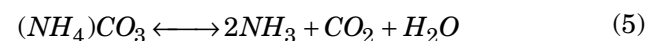$$2NO_2^- + O_2 \xrightarrow{-4e^-} 2NO_3^- \qquad \text{(Nitrobacter)} \qquad (3)$$

Ammonia volatilization is the gaseous loss of NH_3 that occurs when ammonium is surface applied to a calcareous soil or when urea, $(NH_2)_2CO$, is surface applied to any soil. Ammonium surface applied to a calcareous soil results in two chemical steps.
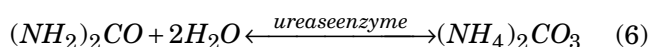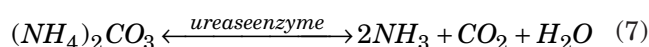
Step 1:

$$CaCO_3 + 2NH_4^+X \longleftrightarrow (NH_4)_2CO_3 + CaX_2 \qquad (4)$$

Step 2:

$$(NH_4)CO_3 \longleftrightarrow 2NH_3 + CO_2 + H_2O \qquad (5)$$

Urea surface applied to any soil results in two chemical steps.

Step 1:

$$(NH_2)_2CO + 2H_2O \xleftrightarrow{urease enzyme} (NH_4)_2CO_3 \qquad (6)$$

Step 2:

$$(NH_4)_2CO_3 \xleftrightarrow{urease enzyme} 2NH_3 + CO_2 + H_2O \qquad (7)$$

Within the formulation, nitrification and ammonia volatilization will be simulated using a combination of methods developed by Reddy *et al.* (1979) and Godwin *et al.* (1984). The total amount of nitrification and ammonia volatilization will be calculated and then partitioned between the two processes. Nitrification is a function of soil temperature and soil water content; ammonia volatilization is a function of soil temperature and depth. Three coefficients will be used in the nitrification/volatilization algorithms to account for the impact of these parameters. It will be assumed that nitrification/volatilization occurs only when the temperature of the soil layer exceeds 5°C. The nitrification/volatilization temperature factor is a function of the soil temperature. The nitrification soil water factor is a function of the soil water content, the amount of water held in the soil layer at wilting point

content, and the amount of water held in the soil layer at field capacity water content. The volatilization depth factor is a function of the depth from the soil surface to the middle of the layer. The impact of environmental factors on nitrification and ammonia volatilization in a given layer is defined by the nitrification regulator and volatilization regulator. The nitrification regulator is a factor of the nitrification/volatilization temperature factor and the nitrification soil water factor. The volatilization regulator is a function of the nitrification/volatilization temperature factor and the volatilization depth factor. The total amount of ammonium lost to nitrification and volatilization is calculated using a first-order kinetic rate equation (Reddy *et al.,* 1979), which is a function of the amount of ammonia, the nitrification regulator, and the volatilization regulator. To partition nitrogen between nitrification and volatilization, the expression by which ammonia is multiplied is solved using each regulator individually to obtain a fraction of ammonium removed by each process. The amount of nitrogen removed from the ammonium pool by nitrification and volatilization is thus a function of these computed fractions.
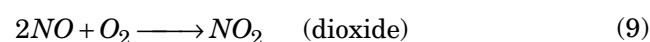
Denitrification is the bacterial reduction of nitrate, NO_3^-, to N_2 or N_2O gases under anaerobic (reduced) conditions. Denitrification is a function of water content, temperature, presence of a carbon source, and nitrate. In general, when the water-filled porosity is greater than 60 percent, denitrification will be observed in a soil. As soil water content increases, anaerobic conditions develop because oxygen diffuses through water 10,000 times slower than through air. Because the rate of oxygen diffusion through water slows as the water temperature increases, temperature will also influence denitrification. In this formulation, the amount of nitrate lost to denitrification is a function of the amount of nitrate, the nutrient cycling temperature factor, and the amount of organic carbon.

Lightning discharge converts atmospheric N_2 to nitric acid, which can then be transferred to the soil with precipitation. The chemical steps involved are
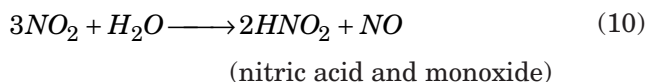
Step 1:

$$N_2 + O_2 \xrightarrow{arc\ of\ electricity} 2NO \qquad \text{(monoxide)} \qquad (8)$$

Step 2:

$$2NO + O_2 \longrightarrow NO_2 \qquad \text{(dioxide)} \qquad (9)$$

Step 3:

$$3NO_2 + H_2O \longrightarrow 2HNO_2 + NO \qquad (10)$$

(nitric acid and monoxide)
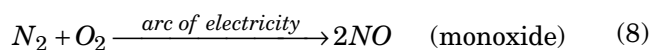
More nitrogen will be added to the soil with rainfall in areas with a high amount of lightning activity than in areas with little lightning. The amount of nitrate added to the soil in rainfall is a function of the concentration of nitrogen in the rain and the amount of precipitation on a given day. The nitrogen in rainfall is added to the nitrate pool in the top 10 mm of soil.

Legumes can supply a portion of their nitrogen demand through fixation of atmospheric N_2 performed by rhizobia living in association with the plants. In exchange for nitrogen, the plant supplies the bacteria with carbohydrates. Within this formulation, nitrogen fixation by legumes is simulated when the soil does not supply the plant with the amount of nitrogen needed for growth. The nitrogen obtained by fixation is incorporated directly into the plant biomass and does not enter the soil unless plant biomass is added to the soil as residue after the plant dies.

As water evaporates from the soil surface, the water content at the surface drops and creates a gradient in the profile. Water from lower in the profile will move upward in response to the gradient, carrying dissolved nutrients with it. Nitrate is transported from the first soil layer to the surface top 10 mm of soil as a function of the nitrate content of the first soil layer, the amount of water removed from the first soil layer as a result of evaporation, and the soil water content of the first soil layer.

Most plant essential nutrients are cations, which are attracted and sorbed to negatively charged soil particles. As plants extract these cations from soil solution, the soil particles release bound cations into soil solution to bring the ratio of nutrients in solution and on soil particles back into equilibrium. In effect, the soil buffers the concentration of cations in solution. In contrast, nitrate is an anion and is not attracted to or sorbed by soil particles. Because retention of nitrate by soils is minimal, nitrate is very susceptible to leaching.

Most soil minerals are negatively charged at normal pH, and the net interaction with anions, such as nitrate, is repulsion from particle surfaces. This repulsion is termed negative adsorption or anion exclusion. Anions are excluded from the area immediately adjacent to mineral surfaces due to preferential attraction of cations to these sites. This process has a direct impact on the transport of anions through the soil, for it effectively excludes anions from the slowest-moving portion of the soil water volume found closest to the charged particle surfaces (Jury *et al.*, 1991). In effect, the net pathway of the anion through the soil is shorter than it would be if all the soil water had to be used (Thomas and McMahon, 1972). Nitrate may be transported with surface flow, lateral flow, or percolation. To calculate the amount of nitrate moved with the water, the concentration of nitrate in the mobile water will be calculated. This concentration will then be multiplied by the volume of water moving in each pathway to obtain the mass of nitrate lost from the soil layer. The concentration of nitrate in the mobile water fraction is a function of the amount of nitrate, the amount of mobile water, the fraction of porosity from which anions are excluded, and the saturation water content of the soil.

Organic nitrogen attached to soil particles may be transported by surface runoff to the main channel. This form of nitrogen is associated with the sediment loading from a grid cell, and changes in sediment loading are reflected in the organic nitrogen loading. The amount of organic nitrogen transported with sediment to the stream is calculated with a loading function developed by McElroy *et al.* (1976) and modified by Williams and Hann (1978). The amount of organic nitrogen transported in the surface runoff is a function of the concentration of organic nitrogen in the soil surface, the sediment yield, and the total area of the computational grid cell.

Phosphorus Cycle

Although plant phosphorus demand is considerably less than nitrogen demand, phosphorus is required for many essential functions. The most important of these is its role in energy storage and transfer. Energy obtained from photosynthesis and metabolism of carbohydrates is stored in phosphorus compounds for later use in growth and reproductive processes.

The three major forms of phosphorus in mineral soils are organic phosphorus associated with humus, insoluble forms of mineral phosphorus, and plant-available phosphorus in soil solution. Phosphorus may be added to the soil by fertilizer, manure, or residue application. Phosphorus is removed from the soil by plant uptake and erosion. Figure 3 shows the major components of the phosphorus cycle.

Unlike nitrogen, which is highly mobile, phosphorus solubility is limited in most environments. Phosphorus combines with other ions to form a number of insoluble compounds that precipitate out of solution. These characteristics contribute to a buildup of phosphorus near the soil surface that is readily available for transport in surface runoff. In addition to precipitating, phosphorus adsorbs to soil solids and can be transported via soil erosion. Sharpley and Syers
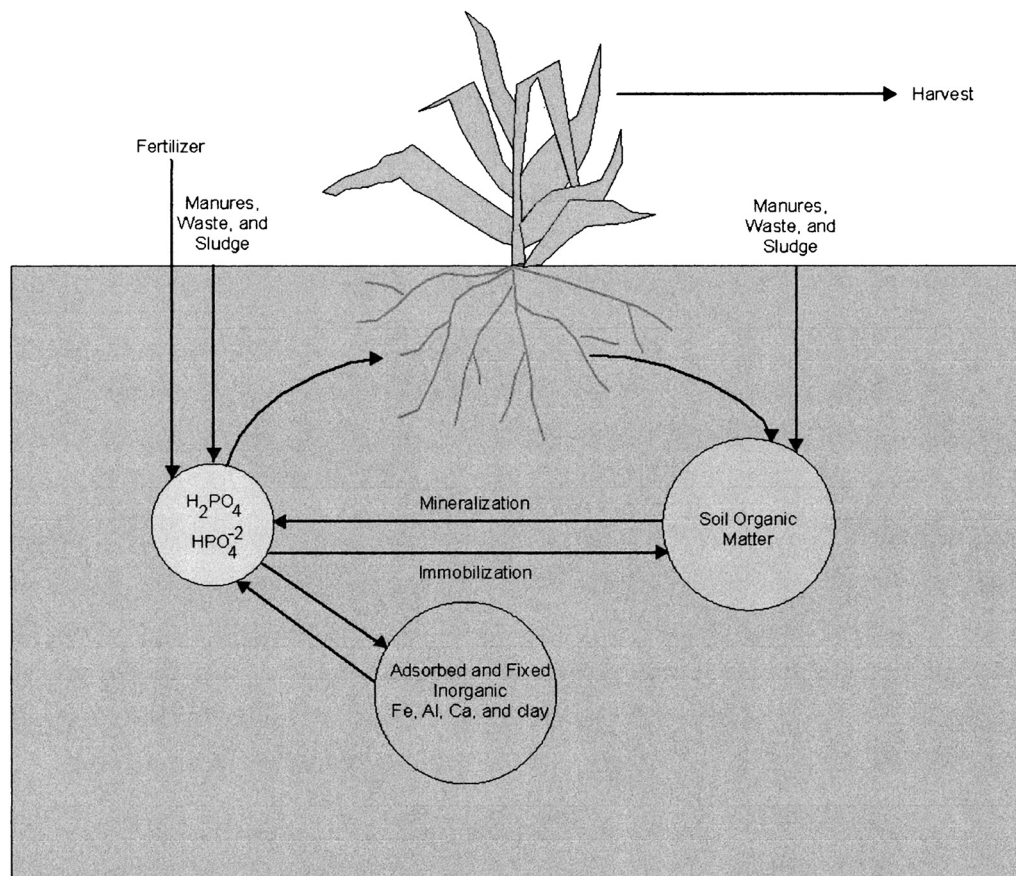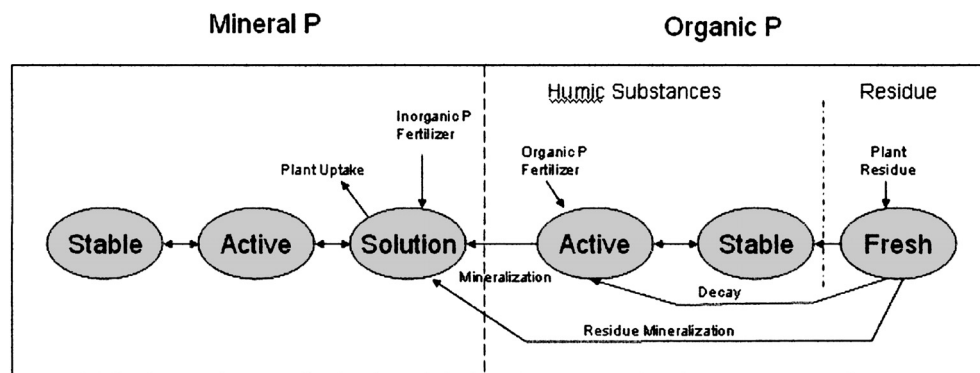
Figure 3. Phosphorus Cycle.

Figure 4. Soil Phosphorus Pools and Processes That Move P In and Out of Pools.

(1979) observed that surface runoff is the primary mechanism by which phosphorus is exported from most catchments.

Six pools of phosphorus are represented in the formulation in Figure 4. Three pools in Figure 4 are inorganic forms of phosphorus, and the other three pools are organic forms of phosphorus. Fresh organic P is associated with crop residue and microbial biomass, and the active and stable organic P pools are associated with the soil humus. The organic P associated with humus is partitioned into two pools to account for the variation in availability of humic substances to mineralization. Soil inorganic P is divided into solution, active, and stable pools. The solution pool is in rapid equilibrium (several days or weeks) with the active pool. The active pool is in slow equilibrium with the stable pool.

The phosphorus mineralization algorithms in this formulation are net mineralization algorithms that incorporate immobilization into the equations. The phosphorus mineralization algorithms developed by Jones *et al.* (1984) are similar in structure to the nitrogen mineralization algorithms. Two sources are considered for mineralization: the fresh organic P pool associated with crop residue and microbial biomass and the active organic P pool associated with soil humus. Mineralization and decomposition are allowed to occur only if the temperature of the soil layer is above 0°C. Mineralization and decomposition are dependent on water availability and temperature. Two factors are used in the mineralization and decomposition equations to account for the impact of temperature and water on these processes.

Phosphorus in the humus fraction is partitioned between the active and stable organic pools using the ratio of humus active organic N to stable organic N. The amount of phosphorus in the active and stable organic pools is a function of the amount of humic phosphorus, the amount of nitrogen in the active organic pool, and the amount of nitrogen in the stable organic pool. Mineralization from the humus active organic P pool is a function of the rate coefficient for mineralization of the humus active organic nutrients, the nutrient cycling temperature factor, the nutrient cycling water factor, and the amount of phosphorus in the active organic pool. Phosphorus mineralized from the humus active organic pool is added to the solution P pool.

Decomposition and mineralization of the fresh organic phosphorus pool is allowed only in the first soil layer. Decomposition and mineralization are controlled by a decay rate constant that is continuously updated.

Many studies have shown that after an application of soluble P fertilizer, solution P concentration decreases rapidly with time due to reaction with the soil. This initial "fast" reaction is followed by a much slower decrease in solution P that may continue for several years (Rajan and Fox, 1972; Barrow and Shaw, 1975; Munns and Fox, 1976; Sharpley, 1982). To account for the initial rapid decrease in solution P, this formulation assumes that a rapid equilibrium exists between solution P and an "active" mineral pool. The subsequent slow reaction is simulated by the slow equilibrium assumed to exist between the "active" and "stable" mineral pools. The algorithms governing movement of inorganic phosphorus between these pools are taken from Jones *et al.* (1984). Equilibrium between solution and the active mineral pool is governed by the phosphorus availability index. This index specifies the fraction of fertilizer P that is in solution after an incubation period (i.e., after the rapid reaction period). The movement of

phosphorus between the solution and active mineral pools will be governed by the equilibrium equations, which are a function of the amount of phosphorus in the active mineral pool, the amount of phosphorus in solution, and the phosphorus availability index. When phosphorus amount is positive, phosphorus is being transferred from solution to the active mineral pool. When the phosphorus amount is negative, phosphorus is being transferred from the active mineral pool to solution. The rate of flow from the active mineral pool to solution is 1/10 the rate of flow from solution to the active mineral pool. This formulation simulates slow phosphorus sorption by assuming that the active mineral phosphorus pool is in slow equilibrium with the stable mineral phosphorus pool. At equilibrium, the stable mineral pool is four times the size of the active mineral pool. When not in equilibrium, the movement of phosphorus between the active and stable mineral pools is a function of the slow equilibrium rate constant, the amount of phosphorus in the active mineral pool, and the amount of phosphorus in the stable mineral pool.

The primary mechanism of phosphorus movement in the soil is by diffusion. Diffusion is the migration of ions over small distances (1 to 2 mm) in the soil solution in response to concentration gradient. The concentration gradient is created when plant roots remove soluble phosphorus from soil solution, depleting solution P in the root zone. Due to the low mobility of phosphorus, this formulation will allow soluble P to leach only from the top 10 mm of soil into the first soil layer. The amount of solution P moving from the top 10 mm into the first soil layer is a function of the amount of phosphorus in solution in the top 10 mm, the amount of water percolating in the first soil layer from the top 10 mm, the bulk density of the top 10 mm (assumed to be equivalent to the bulk density of the first soil layer), the depth of the surface layer (10 mm), and the phosphorus percolation coefficient. The phosphorus percolation coefficient is the ratio of the phosphorus concentration in the surface 10 mm of soil to the concentration of phosphorus in percolate.

Organic and mineral P attached to soil particles may be transported by surface runoff to the main channel. This form of phosphorus is associated with the sediment loadings from the grid cells, and changes in sediment loading will be reflected in the loading of these forms of phosphorus. The concentration of phosphorus attached to sediment in the soil surface layer is a function of the amount of phosphorus in the active and stable mineral pools, the amount of phosphorus in the humus organic pool, the amount of phosphorus in the fresh organic pool, the bulk density of the soil, and the depth of the surface layer (10 mm).

STUDY AREA

The study area selected for testing the nutrient kinetics and transport modules is the 8-Mile Creek Watershed located within the Eau Galle Watershed in Wisconsin (Figure 5). For this case study, only the transport of soluble N and soluble P is presented. As development continues on the plant/soil modules, a more robust case study to evaluate all the kinetic processes will be done. As the plant/soil modules are integrated into GSSHA, it will become necessary not only to estimate the initial soil concentrations, N and P but also to estimate the temporal loadings per grid cell of N and P. The parameter variables necessary to simulate the various processes also will need to be estimated. While this study area is small and instrumented, most watershed applications will not have the observed loadings and runoff that this one possesses. Observations and previous model studies may not be available for use in estimating the necessary process variables needed for the simulations. For all watershed water quality model studies, this is a constraint that is very difficult to overcome without a robust data collection exercise integrated into the project. To develop a predictive watershed water quality model, observed datasets need to be made available or collected as part of the project.
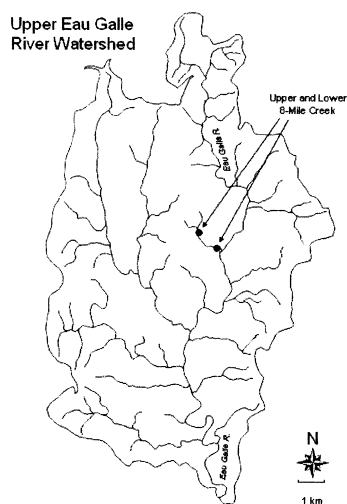
Figure 5. Site Map for 8-Mile Creek Watershed.

The 8-Mile Creek Watershed is approximately 2.3 square kilometers with a maximum elevation of 385 meters National Geodetic Vertical Datum (NGVD) and a minimum elevation of 340 meters NGVD (Figure 6). The watershed is rural in nature, with four main land uses – wooded, pasture, row crop (corn), and dairy, Figure 7. The soil texture is assumed to be

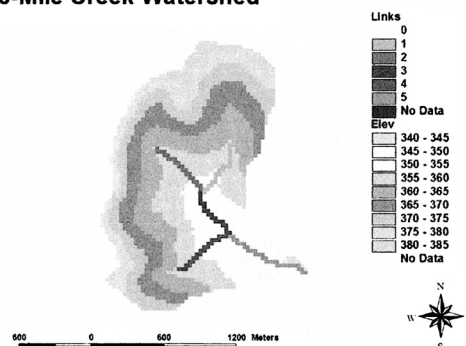uniformly distributed throughout the watershed and is classified as silt loam.
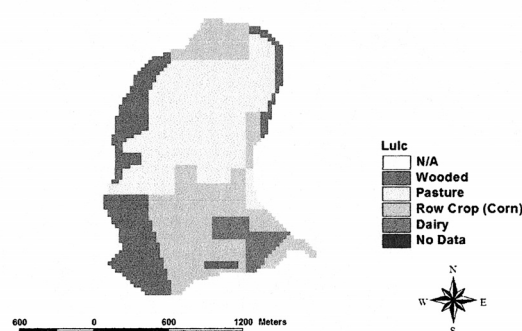

Figure 6. DEM and Channels (meters).

Figure 7. Land Use.

Hydrologic Input Data

Two stream gages and one precipitation gage are used to calibrate the watershed model. The period of record for the stream gages starts on May 8, 2003, and continues to August 1, 2003, at a 15-minute time step (Figures 8 and 9). Precipitation is assumed to be spread uniformly across the watershed. The precipitation period of record starts on April 25, 2003, and continues to October 6, 2003, at a 15-minute time step. The storm event corresponding to June 25, 2003, can be seen in Figure 10.

Nutrient Loading Data

In developing the soluble nitrogen and soluble phosphorus loadings (Table 1), total nitrogen and total phosphorous were measured, and the soluble forms were estimated from these totals. From the total nitrogen (TN) observations, it was assumed that 95 percent of the TN was organic nitrogen. It was also
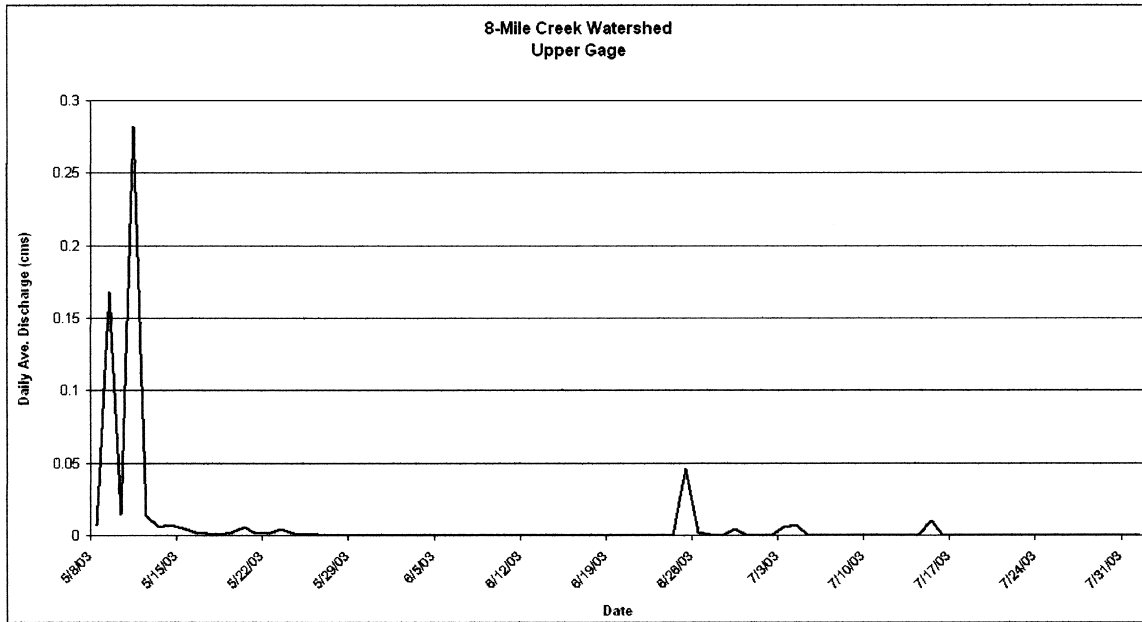
Figure 8. Daily Average Flows (cm) – Upper Gage of 8-Mile Creek Watershed.
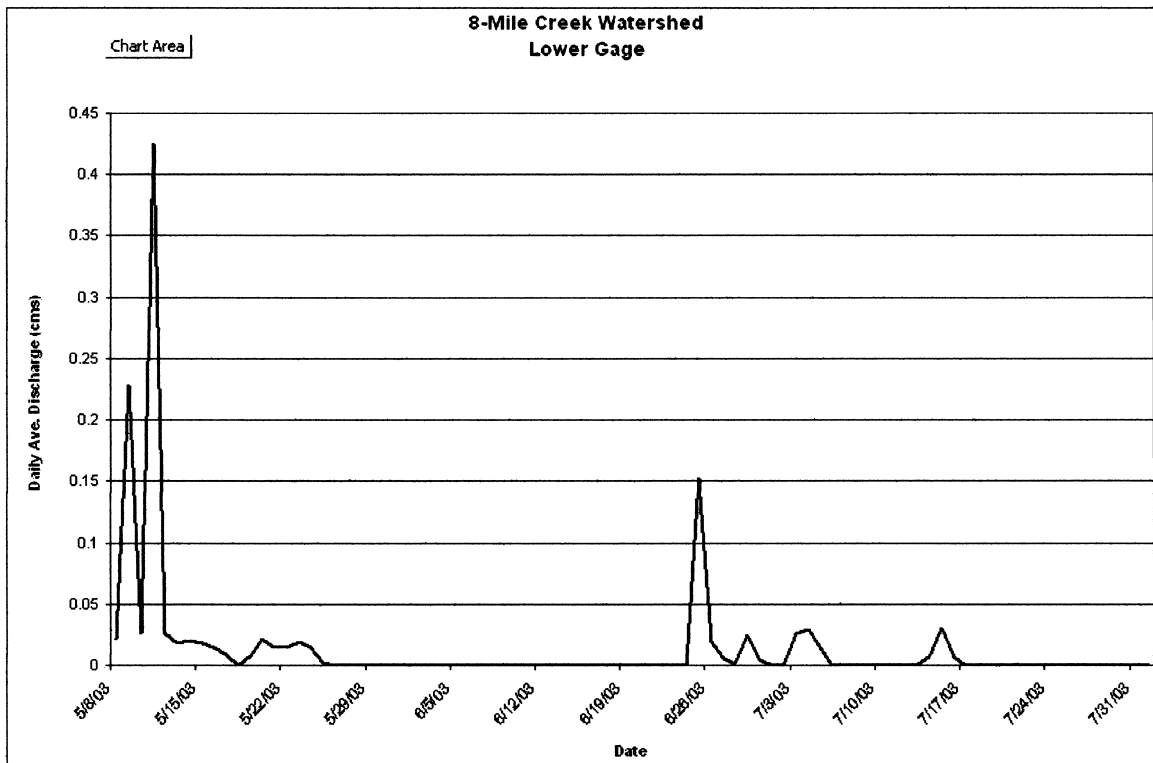

Figure 9. Daily Average Flows (cm) – Lower Gage of 8-Mile Creek Watershed.

assumed that 1 percent of the organic nitrogen would be soluble in the surface runoff and that the effective depth for the soil pore water would be 0.1 cm. The total phosphorus (TP) (Sharpley, 1982) weathered equation was used to compute Labile P (Labile P = 0.56*(Bray Phosphorus)+5.1 in parts per million, ppm). In estimating the soluble P, Kd was assumed to be 175. These estimates were based on engineering
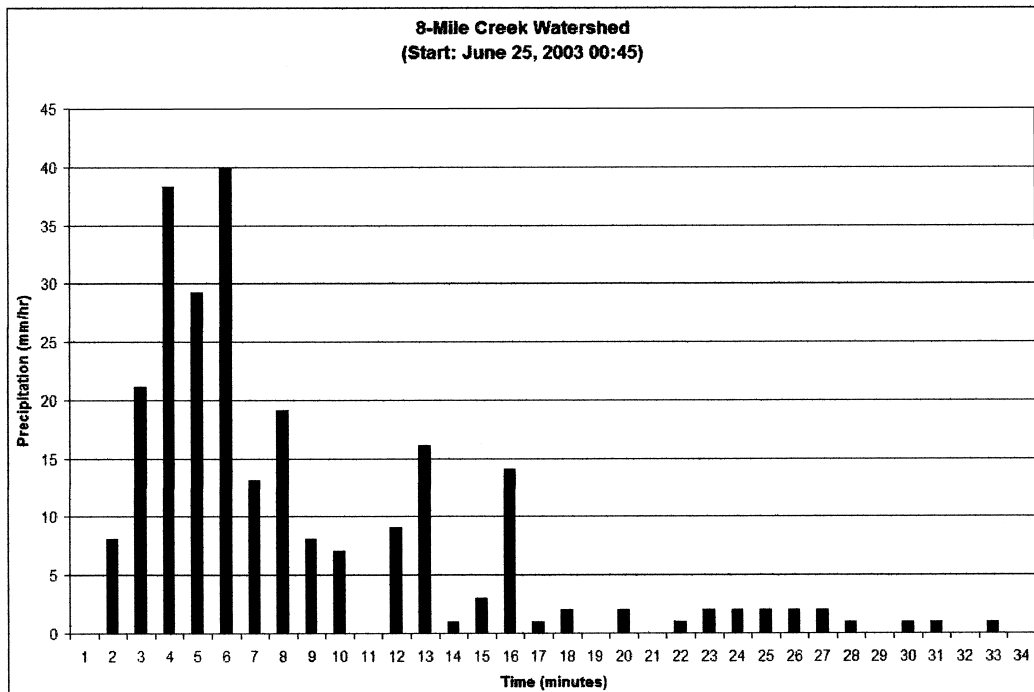
Figure 10. June 25, 2003, Rainfall Hyetograph.

judgment by William F. James, director of the Eau Galle Aquatic Ecology Laboratory, and Terry M. Sobecki, developer of AGNPS and chief of the Environmental Sciences Branch at ERDC.

In developing the soluble nitrogen and soluble phosphorus loadings (Table 1), total nitrogen and total phosphorous were measured, and the soluble forms were estimated from these totals. From the total nitrogen (TN) observations, it was assumed that 95 percent of the TN was organic nitrogen. It was also assumed that 1 percent of the organic nitrogen would be soluble in the surface runoff and that the effective depth for the soil pore water would be 0.1 cm. The total phosphorus (TP) (Sharpley, 1982) weathered equation was used to compute Labile P (Labile P = 0.56*(Bray Phosphorus)+5.1 in parts per million, ppm). In estimating the soluble P, Kd was assumed to be 175. These estimates were based on engineering judgment by William F. James, director of the Eau Galle Aquatic Ecology Laboratory, and Terry M. Sobecki, developer of AGNPS and chief of the Environmental Sciences Branch at ERDC.

Flow Calibration Results

Table 2 shows the comparison between observed and computed flows at both the upper and lower gage sites. Figures 11 and 12 illustrate the results to date for the lower and upper gages of the 8-Mile Creek Watershed.

TABLE 1. Soluble Nitrogen and Soluble Phosphorus Loadings.

Land Use	Soluble N (g)/(grid cell)	Soluble P (g)/(grid cell)
Dairy	192.6	11.10
Pasture	3.6	0.67
Corn	4.1	0.95
Forest	5.1	0.24

TABLE 2. Flow Results at Upper and Lower Gage Sites.

	Runoff Volume (m³)	Percent Difference
Observed Lower Gage	21,074.3	-7.6
Computed Lower Gage	19,463.8	
Observed Upper Gage	11,448.5	-5.6
Computed Upper Gage	10,811.1	

	Time to Peak (min)	Percent Difference
Observed Lower Gage	345	13.0
Computed Lower Gage	390	
Observed Upper Gage	315	-7.9
Computed Upper Gage	290	

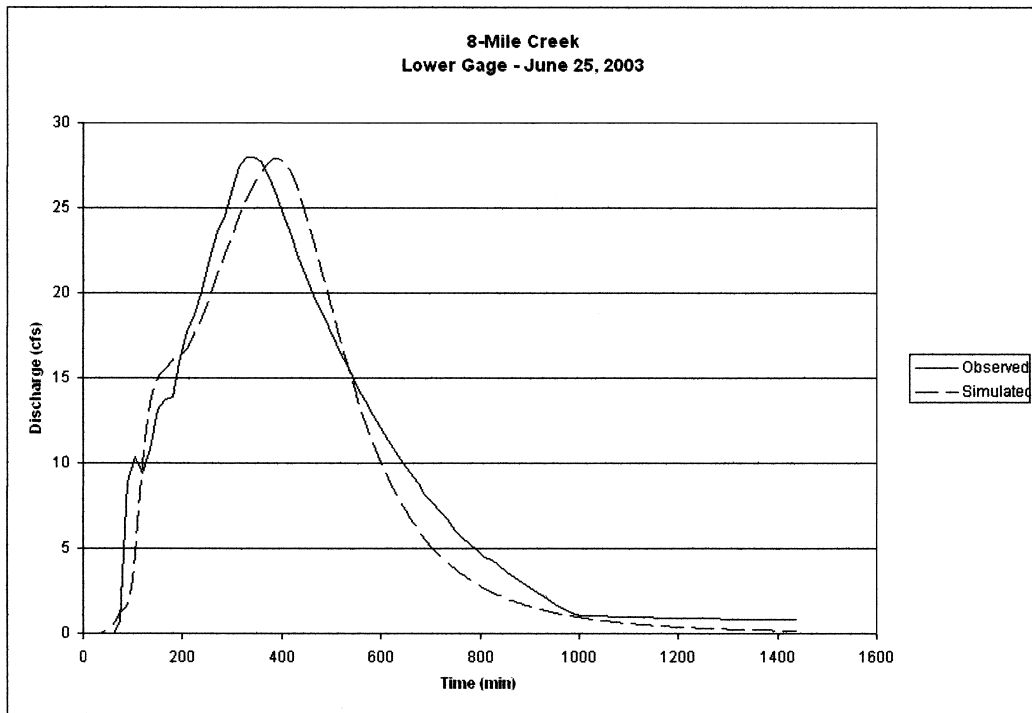|	Peak Flow (cm)	Percent Difference
Observed Lower Gage	0.79	0.0
Computed Lower Gage	0.79	
Observed Upper Gage	0.72	-2.8
Computed Upper Gage	0.70	

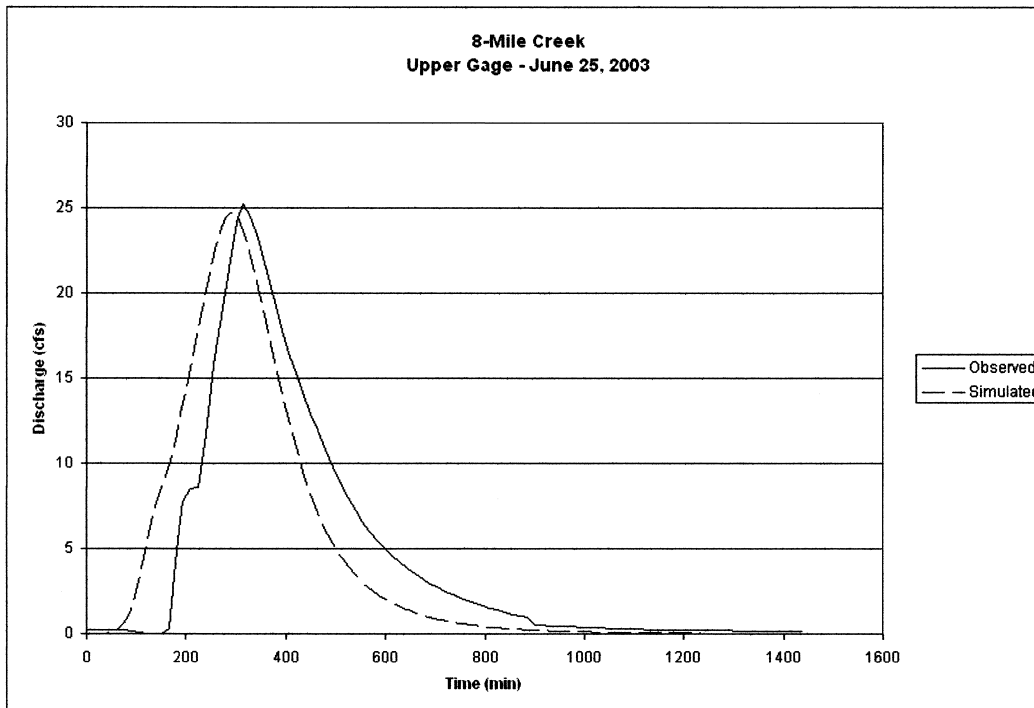Figure 11. Flow Calibration at Lower Gage of the 8-Mile Creek Watershed.

Figure 12. Flow Calibration at Upper Gage of the 8-Mile Creek Watershed.

From these results, the model appears to be capture the catchment response adequately.

Water Quality Calibration Results

Table 3 shows the comparison between total observed and computed soluble N and soluble P for both the lower and the upper gage sites. Figures 13 through 16 show the two-dimensional output for water depth, net sediment erosion and deposition, nitrogen runoff, and phosphorus runoff at a time step of 120 minutes. Figures 17 through 20 show the two-dimensional output for water depth, net sediment erosion and deposition, nitrogen runoff, and phosphorus runoff at a time step of 270 minutes. Figures 21 and 22 show the time series runoff for nitrogen and phosphorous at the lower and upper gage sites.

TABLE 3. Nutrient Results at Upper and Lower Gage Sites.

	Soluble N (kg/d)	Soluble P (kg/d)
Upper Gage Site		
Total Observed	15.0	12.4
Total Computed	20.8	14.9
Lower Gage Site		
Total Observed	163.8	4.0
Total Computed	250.4	3.5

Figures 13 through 20 illustrate the ability to visualize the mass movement of N and P across the landscape and in the channel system. As time progresses, one can see the mass being transported down to the watershed outlet. Figure 21 and Figure 22 show the effects of the increased loading of N and P on the dairy land use in the lower segment of the watershed, which can be seen in the nutrient outflow hydrographs. The lower gage shows a higher peak of mass movement of N and P than does the upper gage, and the timing is earlier, given the high loading close to the watershed outlet.

FUTURE DEVELOPMENT ACTIVITIES

Future development activities will center around improving the water column kinetics and the plant/soil kinetics. Two modeling systems in SWWRP-NSM that will be evaluated and from which algorithms potentially will be taken are the CE-QUAL-RIV1 (RIV1) (Dortch *et al.,* 1995) modeling system and the Ecological DYnamics Simulation (EDYS) (Price *et al.,* 2003) modeling system. RIV1 is a one-dimensional (longitudinal) water quality model for streams. The kinetics found in RIV1 should offer an improvement on the current water-column kinetics found in SWWRP-NSM. The EDYS modeling system is a dynamic vegetation growth model that encompasses natural as well as agricultural land uses. The processes found in EDYS will allow SWWRP-NSM to handle natural land use types in more complexity than is currently available in the SWWRP-NSM's plant/soil kinetic modules.

Water Column Kinetics

The USACE has primary responsibility for regulating U.S. streams, rivers, and waterways. This responsibility often requires resolving water quality issues and concerns within these regulated systems. In an effort to develop an unsteady riverine flow model that handled complex water column water quality kinetics, the ERDC worked with researchers at Ohio State University to develop CE-QUAL-RIV1 (Figure 23). RIV1 is a 1-D model, which implies that vertical temperature, density, and chemical stratifications are nonexistent or negligible for practical purposes. RIV1 was developed for highly unsteady flow systems, thus making it more practical than models such as the U.S. Environmental Protection Agency (EPA) QUAL2E, which relies on steady-state flow computations. Over the years of development, more complex kinetics have been added to RIV1, and thus there is a strong desire to capture those kinetics into the SWWRP-NSM water-column kinetic modules so they can be used by a variety of USACE water engines.

Plant/Soil Kinetics

EDYS has been applied in a wide variety of land and water management scenarios, including: military training, recreational activities, grazing, natural and prescribed burns, fire suppression, eradication, drought assessment, water quality and quantity, reclamation, restoration and revegetation, land cover design, and slope stability.

EDYS (Figure 24) is designed to mechanistically simulate complex ecological dynamics across spatial scales ranging from plots (square meters) to landscape and watershed (square kilometers) levels. Modules include climatic simulators, hydrology, soil profile, nutrient and contaminant cycles, plant community dynamics, herbivory, animal dynamics,
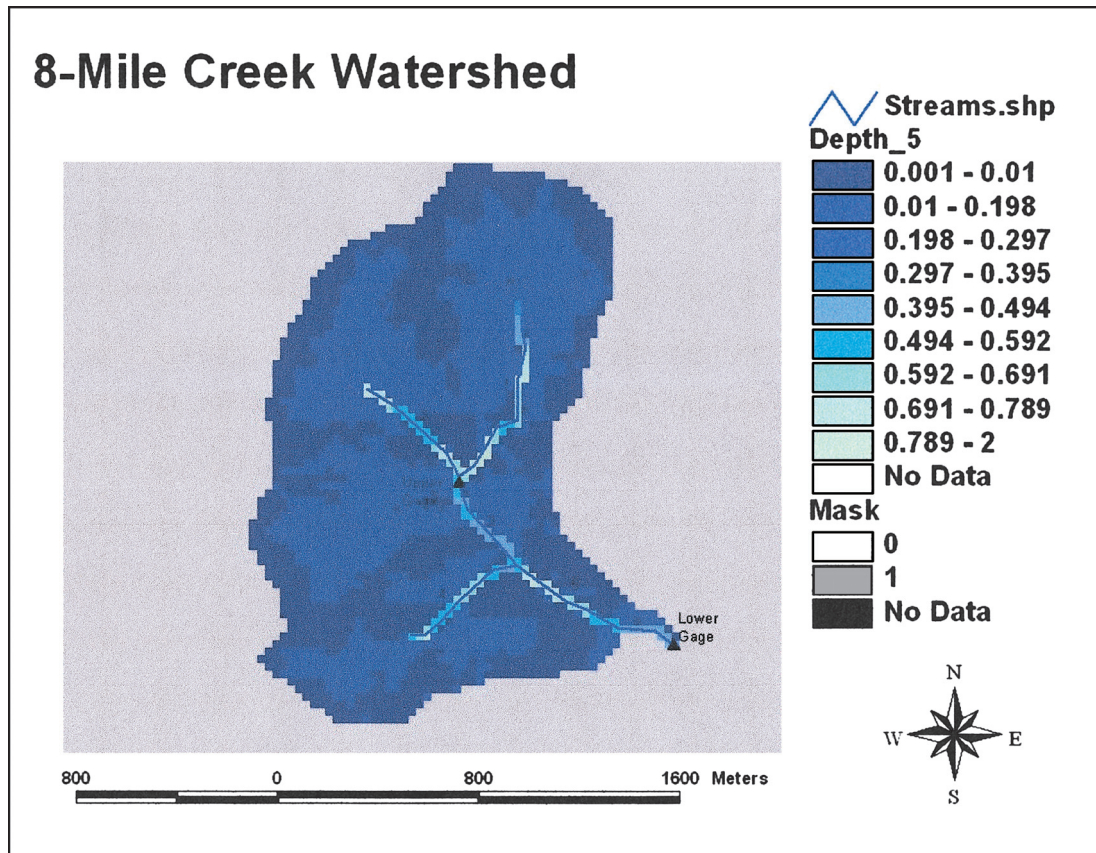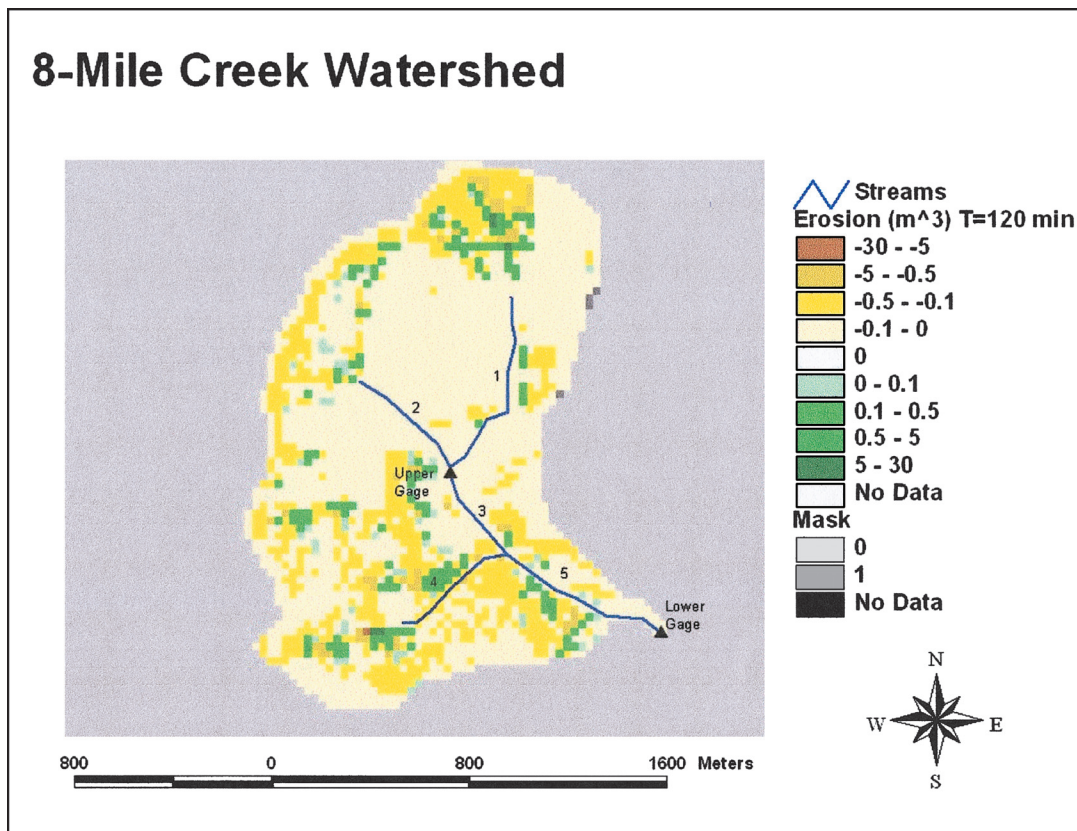
Figure 13. Water Depth (m) at Time = 120 Minutes.

Figure 14. Net Sediment Erosion and Deposition at Time = 120 Minutes.

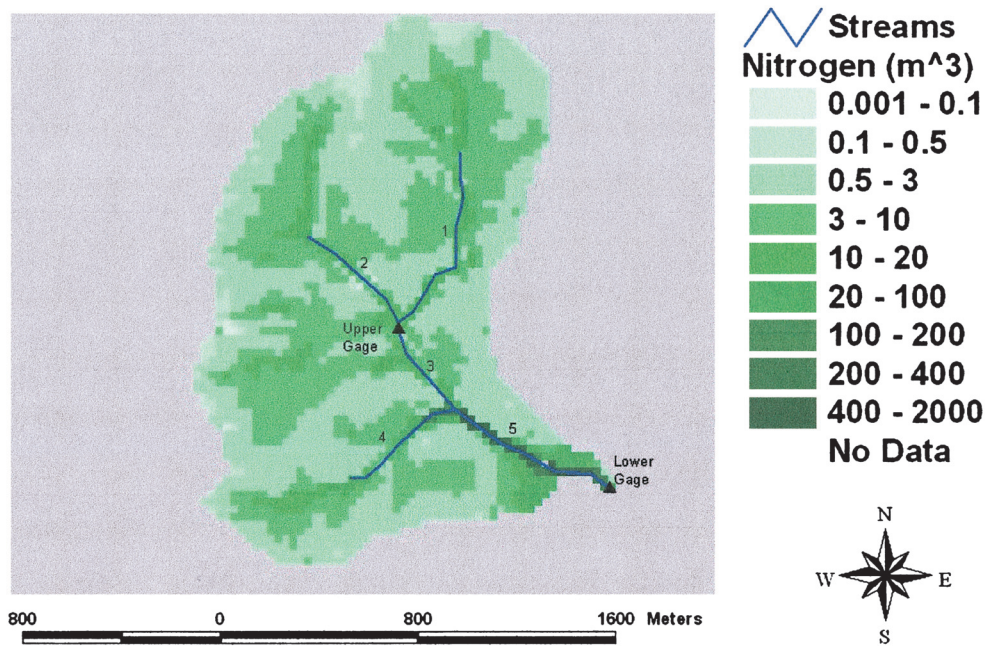Figure 15. Nitrogen at Time = 120 Minutes.
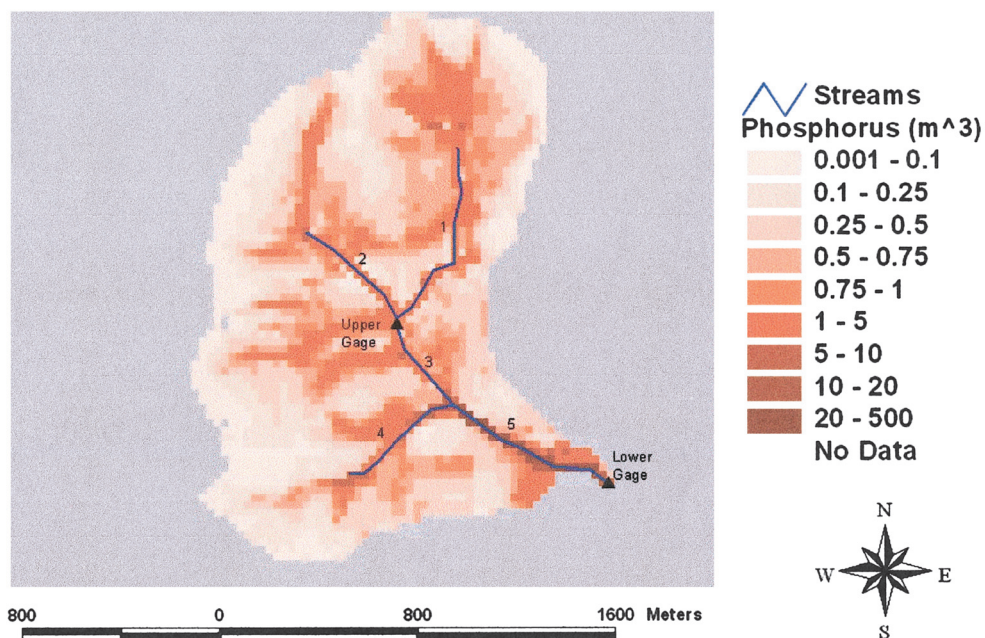
Figure 16. Phosphorus at Time = 120 Minutes.
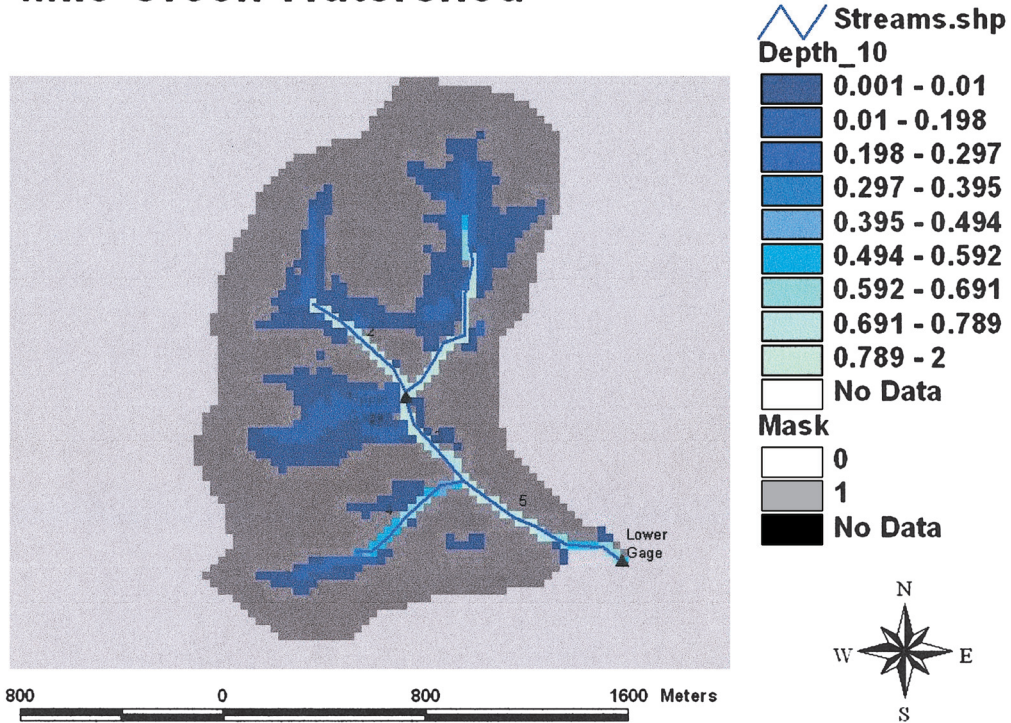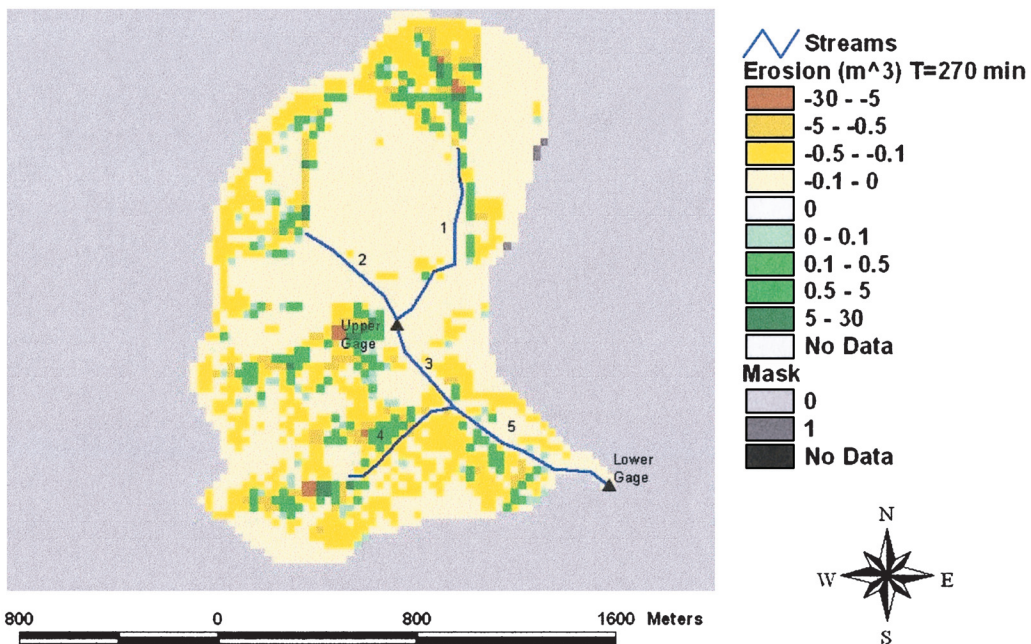
Figure 17. Water Depth (m) at Time = 270 Minutes.

Figure 18. Net Sediment Erosion and Deposition at Time = 270 Minutes.
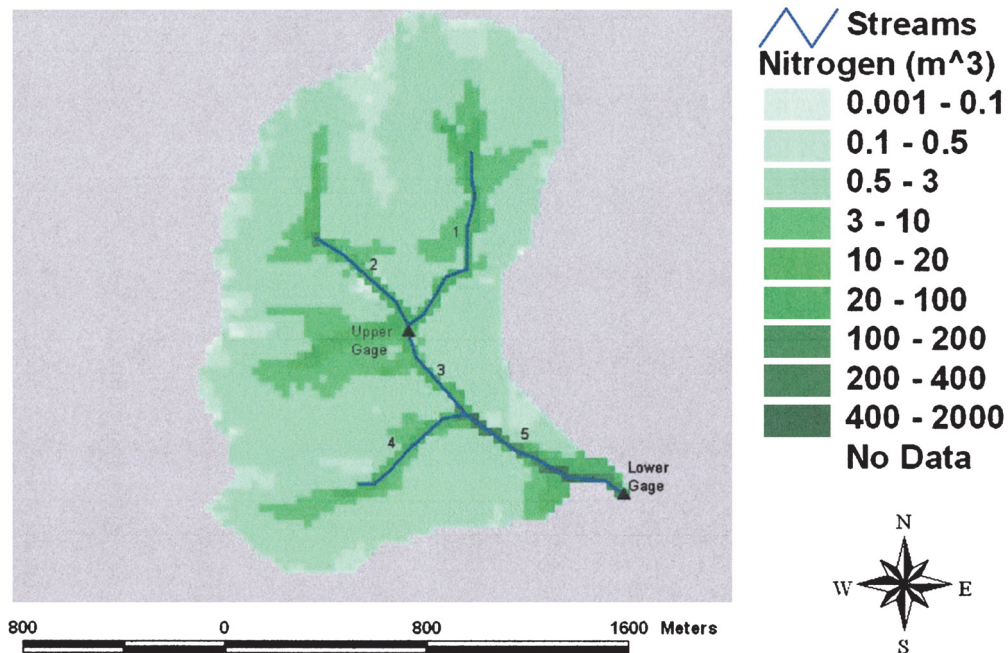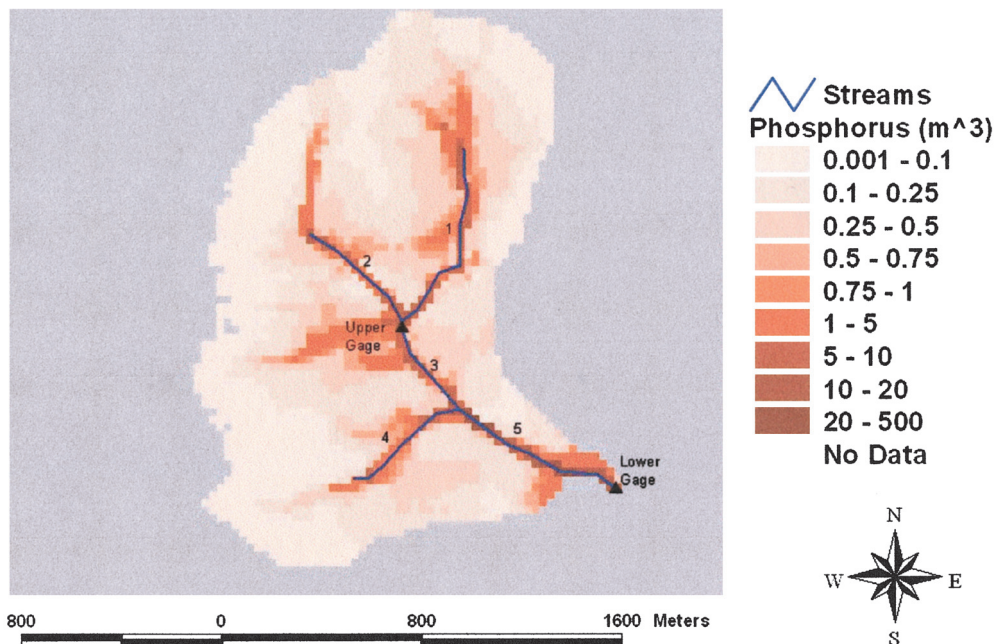
Figure 19. Nitrogen at Time = 270 Minutes.

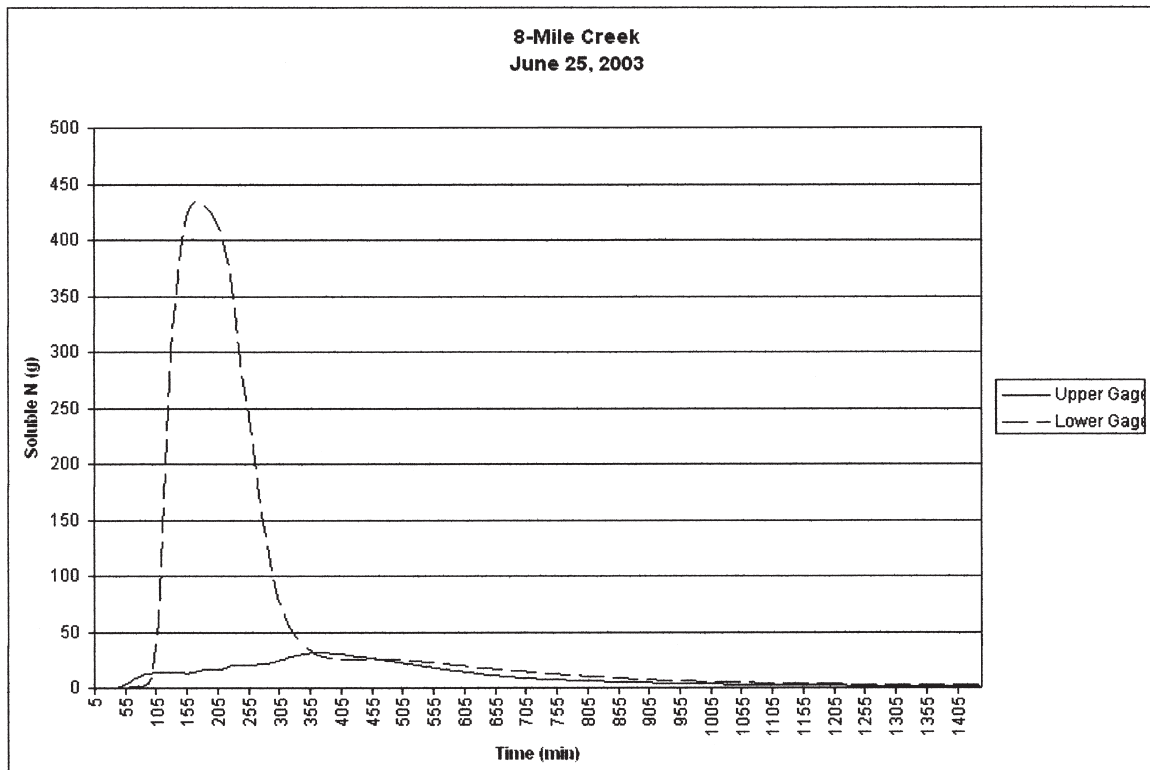Figure 20. Phosphorus at Time = 270 Minutes.
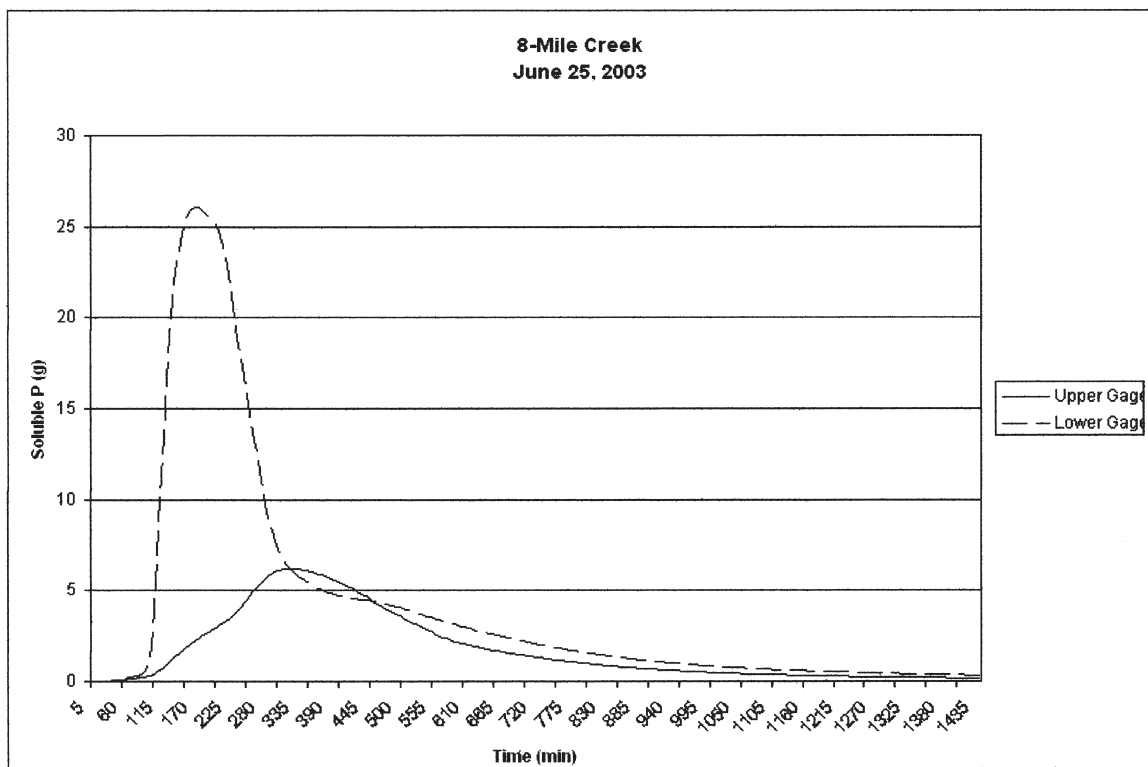
Figure 21. Nitrogen Runoff.
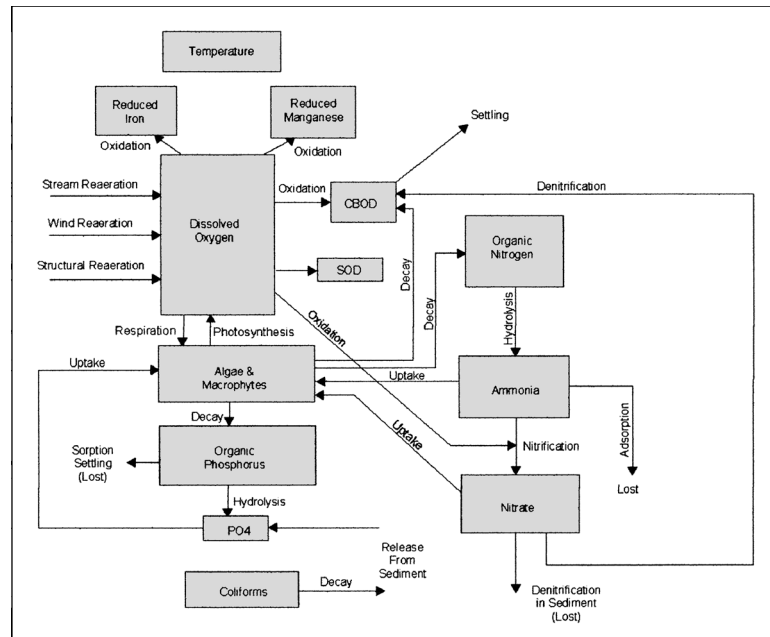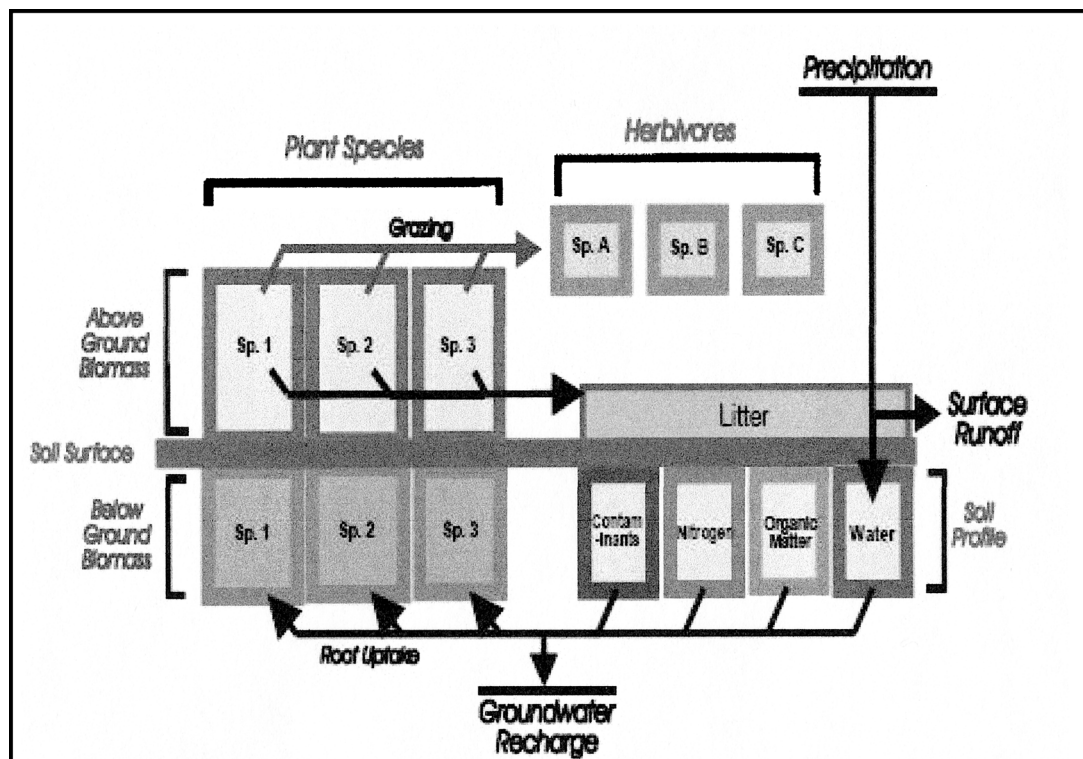

Figure 22. Phosphorus Runoff.

Figure 23. CE-QUAL-RIV1 Modules.

Figure 24. EDYS Modules (Source: Price *et al.,* 2003).

management activities, and natural and anthropogenic disturbances.

While EDYS does account for the hydrologic water budget, it does not route water, sediment, nutrient, and contaminants across the landscape. As larger land areas are considered, erosion, transport, and deposition of sediments, nutrients, and contaminants become more important and will adversely affect the

model results if not properly considered. The implementation of the EDYS process descriptions within the physically-based distributed watershed model GSSHA will provide a state-of-the-art watershed modeling system that will allow for complex hydrologic, sediment, nutrient, and contaminant computations across a dynamically changing landscape.

CONCLUSIONS AND RECOMMENDATIONS

The results from the case study indicate that the transport of soluble constituents across the landscape and through the channel system agrees well with observed measurements. As the plant/soil kinetic modules become available and as the future improvements, EDYS and RIV1 processes, are implemented, SWWRP-NSM will be able to more accurately account for water quality kinetics and thus improve the water quality model results with respect to nutrient and contaminant fate and transport. The integration of SWWP-NSM with the GSSHA model will allow simulation of detailed nutrient processes in the soil, overland, and channel regimes. Since these are simulated on a grid cell basis, the model allows for the transport of material across the overland cells, into the channel cells, and ultimately down to the watershed outlet. Erosion and deposition of material is allowed in the overland and channel regimes, so redistribution of material is allowed so erosional and depositional areas within subareas can be evaluated and compared against field observations. This will lead to better evaluation and design of restoration projects within the watershed.

We recommend that as better understandings of nutrient and contaminant processes become available from field tests, intensive testing needs to be done to make sure they are functioning properly and implemented into the SWWRP-NSM. An extensive literature review needs to be done to ensure that the latest understanding of water quality kinetics are implemented in SWWRP-NSM and that they are as physically-based as is feasible to enhance their use across a wide variety of landscapes. As SWWRP-NSM matures, the modules need to be implemented within selected water engines in such a way as to take advantage of reusable code. This will allow changes to continue to be made within the water quality modules and disseminated to the various modeling systems in an efficient manner, allowing the selected hydraulic and hydrologic modeling systems in the USACE to take advantage of the updates to the water quality kinetic routines. As further testing and evaluation are done, the modeling system needs to be transferred to the public domain so users can apply the model on a wide variety of watersheds and provide feedback to the development team.

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